

PB84-156918

Prediction of Chemical Accumulation by Fish

Argonne National Lab., IL

Prepared for

Environmental Protection Agency, Chicago, IL

Jan 82



U.S. Department of Commerce
National Technical Information Service

NTIS

TECHNICAL REPORT DATA (Please read Instructions on the reverse before completing)		
1. REPORT NO.	2.	3. RECIPIENT'S ACCESSION NO. PB8 4 15691 8
4. TITLE AND SUBTITLE Prediction of Chemical Accumulation by Fish		5. REPORT DATE January 1982-prepared
		6. PERFORMING ORGANIZATION CODE
7. AUTHOR(S) S.A. Spigarelli, M.M. Thommes and A.L. Jensen (School of Natural Resources, U. Michigan, Ann Arbor, MI)		8. PERFORMING ORGANIZATION REPORT NO. ERC 82-09
9. PERFORMING ORGANIZATION NAME AND ADDRESS Ecological Sciences Section Radiological and Environmental Research Division Argonne National Laboratory Argonne, IL 60439		10. PROGRAM ELEMENT NO.
		11. CONTRACT/GRANT NO. EPA-IAG-79-D-F0819
12. SPONSORING AGENCY NAME AND ADDRESS		13. TYPE OF REPORT AND PERIOD COVERED Final 1981
		14. SPONSORING AGENCY CODE
15. SUPPLEMENTARY NOTES		
16. ABSTRACT A bioenergetics model was used to project the accumulation of lipophilic contaminants and to study the factors affecting uptake by Great Lakes fishes. Projections were compared with observed concentrations reported by monitoring agencies. Initial estimates for some model parameters were not applicable to all species, and adjustments of metabolic parameters were necessary to duplicate uptake kinetics among species. The exponent (on weight) for metabolic rate, lipid content, pollutant partitioning and caloric values of consumer and food were parameters that greatly affected the shape and rate of change in uptake curves. Direct uptake from water was insignificant for chemicals with very high log P(octanol/water), compared to uptake from food. Direct uptake from water was more important as log P decreased. Bioaccumulation factors for any of the tested chemicals varied 1-2 orders of magnitude among species within an ecosystem, and typically exceed those predicted by the log BCF-log P regression by 1-2 orders of magnitude.		
17. KEY WORDS AND DOCUMENT ANALYSIS		
a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
Fish Bioaccumulation Chlorinated organics Model-projection metabolism Great Lakes PCBs Dieldrin HCB Mirex Structure-Activity		
18. DISTRIBUTION STATEMENT Release unlimited	19. SECURITY CLASS (This Report) 20. SECURITY CLASS (This page)	21. NO. OF PAGES 44 22. PRICE

PB84-156918

RADIOLOGICAL AND ENVIRONMENTAL RESEARCH DIVISION

ENVIRONMENTAL RESEARCH CONTRIBUTION

No. 82-09

PREDICTION OF CHEMICAL ACCUMULATION BY FISH

By

S. A. Spigarelli, M. M. Thomas, and A. L. Jensen

Since changes may be made before
final publication, this report is
made available with the under-
standing that it will not be used or
reproduced in detail for publication.



ARGONNE NATIONAL LABORATORY, ARGONNE, ILLINOIS

REPRODUCED BY
NATIONAL TECHNICAL
INFORMATION SERVICE
U.S. DEPARTMENT OF COMMERCE
SPRINGFIELD, VA. 22161

PREDICTION OF CHEMICAL ACCUMULATION BY FISH

by

S. A. Spigarelli and M. M. Thommes,
Ecological Sciences Section
Radiological and Environmental Research Division
Argonne National Laboratory
Argonne, Illinois 60439

and

A. L. Jensen
School of Natural Resources
University of Michigan
Ann Arbor, Michigan 48109

A Report To

U.S. Environmental Protection Agency
Great Lakes National Program Office
536 South Clark Street
Chicago, Illinois 60605

Interagency Agreement No. EPA-IAG-79-D-F0819

Project Officer: Vacys J. Saulys
U.S. Environmental Protection Agency
Great Lakes National Program Office

DISCLAIMER

"Although the research described in this article has been funded wholly or in part by the United States Environmental Protection Agency, it has not been subjected to the Agency's required peer and policy review and therefore does not necessarily reflect the views of the Agency and no official endorsement should be inferred."

ABSTRACT

A bioenergetics model, developed by Norstrom et al. (1976) was used to project the accumulation of lipophilic contaminants and to study the factor affecting uptake by fishes in the Great Lakes region. The model projects concentration of chemicals in a population (species) as a function of fish weight or age, and the projections made in this study were compared with observed concentrations reported by state monitoring programs.

Initial attempts to make accurate projections for PCB concentrations in various species and ecosystems met with variable success as initial estimates for metabolic parameters were not applicable to all species. Contrary to conventional theory, we varied the value of the exponent (on weight) for metabolic rate, γ , over the range of 0.8 to 1.30, as this parameter controls the shape of the projected concentration curves. Approximate values of γ were estimated on the basis of lipid content of the species in question, as a working hypothesis. The apparent relationships between γ and lipid content, and between γ and average maximum concentration accumulated indicate that lipid-related metabolism may be the source of variation in uptake among and within species. Asymptotic concentration curves ($\gamma < 1.0$) are common for species with low to intermediate lipid contents while continuously increasing curves ($\gamma > 1.0$) are typical for fishes with high lipid contents. Other model parameters that greatly affected the level of projected uptake are: pollutant concentration in and efficiencies of uptake from food or water; growth-related coefficients; and caloric values of the consumer and its forage.

When applied to the accumulation of PCBs, dieldrin, hexachlorobenzene, and mirex by Great Lakes fishes, the model typically projected concentration curves that were within a factor of two of observed values. In a few cases, the observed data showed distinct differences in uptake between groups of fish

within a population indicating differences in metabolism and/or exposure among individuals. Lipid content of the individuals was highly correlated with the concentrations of each chemical accumulated by a species, but there was no consistent relationship among species, as trophic function (e.g., piscivory versus planktivory) also affected uptake. In general, direct uptake from water was insignificant for chemicals with very high log P (octanol/water) and uptake from food predominated the accumulation process. For chemicals with lower log P (higher solubility in water) direct uptake was more important.

The model was shown to be a useful tool for projecting uptake of lipophilic chemicals by various fish species and the most important parameters in the model have been identified. Parameter estimation is a challenging task as few data are available from the literature, but our experience demonstrates the feasibility of obtaining accurate projections.

Bioaccumulation factors (BAFs) for numerous chemicals in Great Lakes fishes were calculated and empirical relationships with log P and fat content of fishes were developed. BAFs for any chemical vary over 1 to 2 orders of magnitude among species within an ecosystem, and typically exceed that predicted by the log BCF-log P regression by 1-2 orders of magnitude. Species regressions of log BAF versus log P show consistent patterns among species and offer a means of empirical estimation of bioaccumulation by particular species. Regressions of log BAF versus fat content also show consistent trends and offer an alternative method of estimation.

INTRODUCTION

Presently, various methods are being employed to estimate or project the uptake and "equilibrium" concentrations of contaminants that are accumulated by aquatic organisms. Standard laboratory methods for estimating bioconcentration factors (BCF) are subject to biases inherent in the study designs, such as (1) the difficulty in simulating chemical form or speciation as it occurs in nature and (2) the importance of selecting representative species and life-stages for study. Introduction of lipophilic chemicals with solvent carriers, and the use of early life stages in tests usually result in rapid uptake from water and apparent steady-state conditions because of rapid growth (growth dilution of uptake). In natural waters, lipophilic chemicals are preferentially bound to particulate matter, rather than being in solution. Non-polar chemicals with high log P (octanol/water) enter fish primarily via food, rather than via direct uptake from water (e.g., Macek and Korn, 1970; Reinert et al., 1974; Niimi and Cho, 1981; Spigarelli et al., 1982). Thus, BCF (direct uptake) studies with insoluble chemicals are estimating accumulation factors and "steady-state" concentrations by measuring a process that accounts for a small percentage of total uptake. In fact, bioaccumulation factors (BAFs) for lipophilic chemicals in fish tend to be one to two orders of magnitude greater than estimates made with BCF methods (Veith et al., 1979; Spigarelli et al., 1982). For these reasons, the log BCF-log P correlative method (Neely et al., 1974; Veith et al., 1979) should be regarded as a screening tool for identifying biologically-active chemicals, but not as a method for estimating concentrations likely to occur in fishes.

As BAFs for one chemical can vary one to two orders of magnitude among fishes in an ecosystem, accurate projection of uptake at trophic or species levels requires a quantitative understanding of factors controlling uptake and

new conceptual methods for projecting body burdens in fish. Some notable progress has been made using mathematical models to estimate concentrations of chemicals accumulated by fish. Compartment models that ignore growth (Hamelink, 1977) or apply exponential growth (Thomann, 1981) typically predict asymptotic concentration curves over the life-span of a species, but it is apparent that some fishes continue to accumulate some chemicals throughout the life-span (Jensen et al., 1982). Energetics-based uptake models account for age and species-specific growth rates and metabolism of fishes, and better represent the true patterns of uptake (Norstrom et al., 1976; Weininger, 1978; Jensen et al., 1982). These models describe uptake of chemicals in terms of food consumption and respiration, factors that are related metabolically to size and growth. To date, these models have been applied for prediction of PCB and methylmercury uptake by a few species and the results have been very encouraging. Attempts to extend the application to a broad range of species (Jensen et al., 1982) clearly indicated that certain metabolic parameters in the models could not be held constant for all species, as was previously thought.

The major purpose of this study is to test a bioenergetics model (Norstrom et al., 1976; Jensen et al., 1982) under a variety of chemical-species-ecosystem conditions and to determine those parameters in the model which are most critical for accurate prediction. A second objective is to investigate temporary empirical (correlative) methods for estimating bioaccumulation factors for chemicals in nature.

MODEL DESCRIPTION

The model for change in body burden of a contaminant with respect to time (Norstrom et al., 1976) is the mass balance equation

$$\frac{dP}{dt} = e_f C_f R + e_w C_w V - k_u P W^\xi, \quad (1)$$

where

P = body burden of contaminant

e_f = efficiency of toxicant uptake from food

C_f = toxicant concentration in food

R = ration

e_w = efficiency of toxicant uptake from water

C_w = concentration of toxicant in water

V = volume of water passing gills

k_u = clearance coefficient ($g^{-\xi} \cdot wk^{-1}$)

ξ = exponent of body weight (constant).

W = weight of an individual.

This equation states that the rate of change in body burden equals the rate of uptake from the water, $e_w C_w V$, plus the rate of uptake from the food, $e_f C_f R$, minus the rate of clearance, $k_u P W^\xi$. Applying empirical terms for ration and volume of water passing the gills gives:

$$\frac{dP}{dt} = \frac{e_f C_f}{e_{cf}} [\alpha W^\gamma + \beta \frac{dW}{dt}] - k_u P W^\xi + \frac{e_w C_w}{e_{ox} C_{ox} q_{ox}} [\alpha W^\gamma + \beta \frac{dW}{dt}] - k_u P W^\xi, \quad (2)$$

where

α = low routine metabolism parameter ($Kcal \cdot wk^{-1} g^{-\gamma}$)

γ = exponent of body weight

β = coefficient relating growth to energy associated with growth

e_{ox} = efficiency of oxygen uptake

C_{ox} = concentration of oxygen in the water (g/q)

q_{ox} = caloric equivalent of oxygen (kcal·g⁻¹)

e_{cf} = efficiency of assimilation of metabolizable energy from food.

To apply the uptake model, expressions for weight (w) and growth rate (dW/dt) are necessary. Growth of fish was modelled using the von Bertalanfy equation

$$\frac{dW}{dt} = bKW_{\infty} e^{-Kx} [1 - e^{-Kx}]^{b-1} \quad (3)$$

$$W_t = W_{\infty} [1 - e^{-Kx}]^b$$

where

W_{∞} = asymptotic weight

K = growth coefficient

b = parameter of length-weight equation ($W = aL^b$)

The growth equation was applied together with equation (2) to predict uptake of chemicals as a function of fish weight or age.

Data requirements for a model prediction include: (1) length-weight-age-lipid data for the species in question; (2) concentrations of the contaminant in water and food of the species; (3) estimates of the rates of uptake from food and water (obtainable from BCF studies); and (4) estimates for metabolic parameters in the model. In some cases, sufficient growth data were not available from monitoring agencies and growth parameters were estimated using data from Carlander (1969, 1977). Monitoring data on concentrations of contaminants in fish were used as "ground truth" to evaluate the accuracy of

predictions with the model. A major difficulty of this study was the acquisition of appropriate monitoring data, as most surveillance programs are focused on trend detection rather than species-specific variations in uptake. Thus, most field data sets are restricted to narrow size ranges for a species and to few species. Despite these problems, a total of 22 test cases were modelled (Table 1) covering a variety of species and contaminants in the Great Lakes region.

The procedure for a modelling run consisted of the following steps: (1) acquisition of an appropriate data set (e.g., PCBs in lake trout from Lake Michigan); (2) estimation of growth-related parameters using monitoring data, if provided, or published length-weight data (Carlander, 1969, 1977); (3) estimation of concentrations of the chemical in water and food of the fish species; (4) estimation of uptake and elimination coefficients for the fish-chemical case; (5) estimation of metabolic parameters (initial estimates from Norstrom et al., 1976); and (6) adjustment of parameter values, if necessary, to provide the best fit to the observed (monitoring) data.

PCB ACCUMULATION

Lake Michigan Fishes-PCBs

Initial applications of the model using parameter estimates given by Norstrom et al. (1976; Table 2) resulted in variable success. Preliminary model applications gave projected PCB uptake curves that were asymptotic at high fish weights, while field (monitoring) data for some species showed increases in concentrations over the entire weight/age range (Jensen et al., 1982). As the model was developed and applied for yellow perch, it was not surprising that some parameters would have to be adjusted for species-specific differences in metabolism. Sensitivity analyses indicated that the shape of

the concentration-weight curve was predominantly affected by the exponent on weight for metabolic rate (γ) and that values of $\gamma < 1$ gave asymptotic uptake curves, while values of $\gamma > 1$ gave projections that increased with increasing weight/age. Since conventional theory has long held that γ (slope of metabolic rate-weight relationship) is relatively invariable among fish species (~ 0.80) there was no a priori basis from which to estimate γ for a given species. Therefore, γ was adjusted within the range of 0.8 to 1.30 (a range of values often measured in experiments, but discounted for various reasons; Paloheimo and Dickie, 1966; Glass, 1969) until the projected concentration-weight curve approximated the trend in the field data.

The model was tested with ten PCB data sets for Lake Michigan (Table 1). In each case, uptake and elimination parameters and PCB concentration in Lake Michigan water were held constant, while growth and metabolic parameters and PCB concentration in food were adjusted, as appropriate for each species. For lake trout and salmon (Figures 1-4) the observed PCB concentrations increased with fish weight, and model predictions using $\gamma = 1.0$ to 1.20 resulted in excellent fits to the observed data. For other piscivores, such as brown trout, burbot and northern pike (Figures 5-7) the observed PCB concentrations attained asymptotic levels at high fish weights, and model projections with $\gamma < 1$ resulted in good fits to the observed data. For other fishes, such as whitefish and carp (both omnivorous, but not piscivorous) observed PCB concentrations increased over the weight range (Figures 8-10) and model projections with γ values of 1.20 and 1.30, respectively, approximated trends in the observed data.

These ten tests of the model indicated that trophic position of a species has little bearing on the shape of the PCB concentration-weight curve or on the maximum concentrations accumulated. Northern pike and lake trout are both

terminal piscivores which feed heavily on alewife, but there is an order of magnitude difference in the concentrations of PCBs accumulated by these species. Although carp are terminal consumers in a relatively short food chain (detritus-benthos-carp) levels of PCBs in carp are the highest of all fishes in Lake Michigan. As a working hypothesis, we theorized that species-specific differences in metabolism, and therefore uptake of PCBs, may be a function of differences in lipid content, since the species with asymptotic uptake curves and relatively low BAFs tend to be those with low lipid content. In the absence of any other basis for estimating gamma, we assumed that γ varied directly with lipid content and applied appropriate values for each model run.

As a test of this hypothesis, we applied the results of an experiment on temperature effects on growth and uptake of PCBs by brown trout (Spigarelli et al., 1982) to the model by initializing conditions (metabolic parameter estimates) to those observed in the experiment, and projected the accumulation of PCBs under these conditions (Figure 11). The model projections clearly show that increased growth rate and lipid content significantly increased the uptake of PCBs by brown trout. The projected concentration curves bracketed the observed concentrations and indicated that shifts in metabolism and growth could account for the variability in observed PCB concentrations in brown trout from Lake Michigan.

Lake Ontario-PCBs

PCB accumulation by brown trout and smallmouth bass in Lake Ontario was modelled, using adjusted estimates for (1) PCB concentrations in water and food; and (2) age-growth parameters. Both species have relatively low lipid contents, and model projections using gamma values of 1.0 and 0.90, respectively, provided good fits to the observed data (Figures 12-13). For

brown trout, any single projection did not fit all of the observed data, as the largest fish in the observed data had very high concentrations of PCBs, relative to the smaller fish. By increasing gamma above 1.0 or by increasing the estimate for C_f (PCBs in food) the projection could be forced through the observed values at high weights, but the values at low weights were overestimated. By applying different lipid values or different C_f values between low and high fish weights, the projection was made to fit all of the observed data. This indicates that brown trout in Lake Ontario change diets (PCB and lipid content of food) or experience size-related changes in metabolism over their life-span.

For smallmouth bass, observed PCB concentrations were rather high, even though this species is rather low in fat content. Assuming that γ should be <1 (low fat content) the model projection fit the data only if C_f was increased to 4 ppm (compared to 2 ppm for brown trout). Since smallmouth bass are known to select crayfish as forage, it is conceivable that this benthic organism contains high concentrations of PCBs, but no field measurements on crayfish could be found.

Other PCB Data Sets

Adequate field data sets were available for (1) carp from the Mississippi River (Figure 14); (2) channel catfish from Lake St. Clair (Figure 15); and (3) lake trout from Cayuga Lake, New York (Figure 16). Observed PCB concentrations for carp indicated two groups of fish in the sample, differing in their fat-weight relationships. There is no obvious explanation for this difference, but we were able to project the observed divergence by adjusting γ and lipid values.

The projection for channel catfish was straightforward and provided an excellent fit to the observed data. Although the field data did not include

lipid values, other sources show channel catfish to be intermediate in fat content, and a gamma value of 1.0 gave a good fit to the observed data.

PCB accumulation by lake trout in Cayuga Lake was modelled using metabolic parameter values estimated for lake trout in Lake Michigan. The projected concentration curve was an excellent fit to the observed data.

ACCUMULATION OF OTHER CONTAMINANTS

A rigorous test of any bioaccumulation model should include a range of chemicals that are or may be of concern as extant contaminants. The choice of chemicals to be modelled was severely limited by the availability of analyses performed by monitoring agencies. Good data sets were obtained for dieldrin, hexachlorobenzene (HCB) and mirex, chemicals that span a log P range of ~4.4 to 6.9, and, therefore, provide a rigorous test of the model. Unfortunately, much less is known about the rates of uptake and elimination and the concentrations of these chemicals in natural waters/biota, compared to PCBs. In some cases, our parameter estimates are "guesstimates" as no information was available to allow parameter estimation.

Dieldrin

Dieldrin concentrations in lake trout from Lake Michigan (Figure 17) show a large variation at low weights and less variation at high weights, indicating different exposures and/or metabolism among individual fish during young stages. Any single projection with the model fit one weight group or the other, but not both. By adjusting the concentration of dieldrin in water (C_w) between weight groups, a combined projection was made to fit all of the observed data.

American eel from Lake Ontario also show highly variable dieldrin concentrations at low weights (Figure 18) but a single projection with the

model provided an adequate fit to the observed data.

Hexachlorobenzene (HCB)

Although concentrations of HCB are quite low in Great Lakes fishes, compared to PCBs, the model was useful for accurate prediction. For brown trout in Lake Ontario (Figure 19) there is considerable scatter in HCB concentrations, but a single projection approximated the average HCB values observed over the weight range. Concentrations of HCB in American eel from Lake Ontario (Figure 20) were approximately equal to those in brown trout, despite the large differences in fat content between these species (brown trout, 0-15% fat; eel, 3-35% fat). The projection of HCB concentrations in eel was adequate, but different parameter estimates may provide a better fit.

Mirex

Concentrations of mirex in brown trout (Figure 21) and smallmouth bass (Figure 22) from Lake Ontario were readily projected with the model, but in the case of smallmouth bass, the monitoring data indicated two groups of fish. It is not clear what the source of this variation may be, as location and date of collection were not contributing factors. Food source (concentrations of mirex in food) may be the factor causing the two-fold difference between bass groups, as a two-fold change in C_f provided projected concentration curves that fit both groups.

CONCLUSIONS AND SYNTHESIS OF RESULTS

As bioenergetics models have only recently been applied to the prediction of contaminant bioaccumulation, experience with the models was limited to a few species and chemicals. Our applications of the model developed by Norstrom et al. (1976) to a large number of species and more chemicals demonstrated the general utility of the model, and indicated the critical

importance of metabolic and growth parameters.

Influence of Lipids

The effects of lipid content on concentrations of lipophilic chemicals accumulated by fish have been reported many times in the literature. In many reports, a direct relationship was found within or among the species tested, and it has become a "rule of thumb" to expect relatively high concentrations of lipophilic contaminants in fishes with high lipid contents. Our study has provided some interesting perspectives on this concept. In every case where monitoring data included lipid values, we performed linear regressions of concentration versus lipid content. With only one exception (PCBs versus lipid in burbot) the concentration of a contaminant increased linearly with increased lipid content, but in no case did variations in lipid "explain" more than 62% ($R^2 = 0.62$) of the variation in contaminants among individuals. In most species, lipid content increases with time (size/age), and the two independent variables (lipid and time) are highly correlated within each species.

Lipid contents of the fish species in question (q_f) and of its major food (q_{fd}) are important parameters in the model, especially if these values differ significantly. In some model test cases (e.g., carp), the projected concentrations fit the observed data only if lipid-related parameters were adjusted to match the ratios expected in nature, while in other cases (e.g., trout and salmon), the ratios of lipid in fish to those in food approximate unity and do not affect the projections greatly. Part of the "error" in projection of contaminant concentrations over the life span of various fishes may be a result of age-related changes in lipid content that are not included in the model.

Route of Uptake

The direct uptake of contaminants from water by fish has been the focus of most studies, and the BCF methods are based on the assumption that this process is important over a wide range of chemical properties. Recently, however, there is mounting evidence from experimental and modelling studies that uptake from food may predominate, at least with high log P chemicals. We used the model to estimate the proportions of uptake from food and water for each chemical-species case (Table 3). In general, uptake of dieldrin from water predominated over uptake from food for lake trout/Lake Michigan and American eel/Lake Ontario. Uptake of PCBs and mirex from food predominated over uptake from water in all species tested, while both routes of uptake may be important for HCB. These results indicate a log P effect on route of uptake in fishes, with low log P (higher solubility in water) favoring the direct route and low solubility (high log P) favoring the trophic route. However, the variability in percentage uptake of PCBs from water among fishes (0.9-17%) indicates that species- and ecosystem-related differences in route of transfer are large and that solubility is not the only factor controlling the route of uptake by fishes. Percentage uptake of PCBs from Lake Michigan water was highest in fishes with the lowest lipid content (Northern pike and burbot), but there was no significant linear relationship between percentage uptake from water and average lipid content among fishes within an ecosystem.

Comments on the Model

The previous discussion of modelling results clearly demonstrates that the bioenergetics model developed by Norstrom et al. (1976) is a valuable tool for projecting the concentrations of various chemicals that are accumulated by fish. As with any mathematical model, the accuracy of the projection is highly dependent on the accuracy of parameter estimates. Presently, only

limited information is available from the literature and monitoring agencies for many of the extant contaminants in nature, and, therefore, parameter estimation is often not based on measurements, but on the best guesses of the researcher. This can be a problem, particularly if the most critical parameters are inaccurately estimated.

Our experience with the model has shown that the most critical parameters (that is, those greatly affecting the shape and/or magnitude of the projected concentrations) are: (1) γ (the slope of the metabolism-weight relationship); (2) C_f or C_w (concentration in food and water); (3) e_f or e_w (efficiencies of uptake from food and water); (4) K_u (the clearance coefficient); (5) K (the growth coefficient); and (6) q_f and q_{fd} (the caloric equivalents of the species in question and its major foods). Clearly, γ and C_f or C_w are the parameters requiring the most accurate estimation, assuming other parameter estimates approximate the true values. Generally speaking, if gamma and concentrations in food/water are accurately estimated, the projected concentration curve is within a factor of 2-4 of the observed values. As concentrations of any chemical vary by more than a factor of 10 among species within a particular ecosystem, a projection within a factor of 2-4 is a significant improvement over present methods, such as the BCF-log P correlation.

Thus, we have demonstrated the utility of the model for estimating/projecting the concentrations of chemicals accumulated by fish. In cases where particular pollutants are perceived to be problems (site-specific or ubiquitous), the model is very useful for estimating changes in uptake, given changes in availability (inputs, concentration in water, food, etc.). Various other applications for assessment of contaminant effects are possible, such as prediction of biological activity of new chemicals. A second, general

area of application for the model is its use as a research tool. Through sensitivity analyses, the model provides a framework for identifying the most important mechanisms in the bioaccumulation process. Clearly, these mechanisms are not well understood in a general sense, and less so in specific cases, as they change in importance with changes in properties of chemicals and biota of interest.

BIOACCUMULATION FACTORS IN GREAT LAKES FISHES

A second major purpose of this study was to estimate actual bioaccumulation factors in Great Lakes fishes and to investigate the potential for developing empirical methods of predicting BAFs. In addition to the monitoring data sets used for modelling tests, some contaminant data were available for other species, and average maximum concentrations of PCBs, pp'DDT, dieldrin, chlordane, heptachlor, toxaphene and HCB were estimated for species that had been analyzed. Concentrations of these chemicals in water and fishes of Lakes Michigan and Ontario, and BAFs were estimated using information from various published and unpublished sources (Tables 4 and 5).

Plots of the estimated BAFs for chemicals in fishes from Lakes Michigan and Ontario are given in Figures 23 and 24, along with the log BCF-log P regression (Veith et al., 1979). Inspection of these figures indicates the following: (1) the log BCF-log P equation predicts concentration factors that are 1-2 orders of magnitude lower than observed BAFs, at least within the log P range of 4.4 to 6.9; (2) the range of BAFs for any single chemical among species is about 1 to 1.5 orders of magnitude; and (3) BAFs tend to fall in about the same species order for each chemical. In Lake Michigan, lake trout and carp (long-lived, fatty species) showed the highest log BAFs, which approximate 7.0 for chemicals with log P >5.8. Other salmonids have slightly

lower BAFs, followed by other species that are less piscivorous, more planktivorous or have lower lipid content. A similar range and species-order of BAFs is seen for Lake Ontario fishes.

It is apparent from the above that a single regression between BAFs and log P will provide order of magnitude accuracy in estimation of BAF for a given species-chemical combination, compared to the much lower accuracy of the BCF-log P regression. Because assessments of chemical impacts usually focus on selected species that are of social or biological importance, we developed species-specific regressions for cases where ample data were available (Table 6). In general species regressions showed significant positive slopes and very good linear fits (R^2). Exceptions were species that function as planktivores or benthivores (bloaters, whitefish, yellow perch, burbot, alewife). As the Lake Ontario data spanned a wider range of log P, the slopes for given species are not strictly comparable between lakes, and the Lake Ontario regressions show a tendency for decreasing BAF at high log P (6.9 = mirex) for planktivores (alewife and yellow perch) and piscivores with low fat content (walleye). Examples of significant species regressions are shown in Figures 25 and 26.

An alternative method of predicting species-specific bioaccumulation factors, based on lipid content, was investigated by examining the relationship between average maximum lipid content (species) and log BAF for each chemical (Table 7). In cases where the degrees of freedom were high (>8), significant linear regressions were demonstrated, but there was considerable scatter in observed values around the predicted and the maximum coefficient of determination was 0.68. Thus, for any given chemical (log P), there may be a useful relationship between lipid content of species and their bioaccumulation factors for the chemical (Figures 27 and 28), but predictions

based on this method may deviate from the observed values by about 0.5 log BAF. This source of inaccuracy may result from the influence of factors other than lipid content (such as trophic position and life span) or may reflect inaccuracies in estimation of average maximum BAF and lipid values for each species.

ACKNOWLEDGEMENTS

We thank the Wisconsin and Michigan Departments of Natural Resources and the New York Department of Environmental Conservation for providing published and unpublished data; and the Great Lakes National Program Office, U.S. Environmental Protection Agency, for making this study possible. Ms. Pat Tyrolt typed the manuscript.

REFERENCES

- Haime, C. A., J. W. Serum, W. D. Youngs, and D. J. Lisk. 1972. Polychlorinated biphenyl residues: Accumulation in Cayuga Lake trout with age. *Science* 177: 1191-1192.
- Carlander, K. D. 1969. Handbook of freshwater fishery biology. Volume I. Iowa State University Press, Ames, Iowa. 752 pp.
- Carlander, K. D. 1977. Handbook of freshwater fishery biology. Volume II. Iowa State University Press, Ames, Iowa. 431 pp.
- Daly, R., J. Moore, and P. Schultz. 1975. Wisconsin Department of Natural Resources, Report to the Great Lakes Fishery Commission. Wisconsin Department of Natural Resources, Madison, Wisconsin. 28 pp.
- Glass, N. R. 1969. Discussion of calculation of power function with special reference to respiratory metabolism in fish. *J. Fish. Res. Board Can.* 26: 2643-2650.
- Great Lakes Environmental Contaminants Survey. 1975. Data reports. Michigan Department of Agriculture, Lansing, Michigan.
- Hamelink, J. L. 1977. Fish and chemicals: The process of accumulation. *Ann. Rev. Pharmacol. Toxicol.* 17: 167-177.
- International Joint Commission. 1978. Status report on the persistent toxic pollutants in the Lake Ontario basin. Report to the Great Lakes Water Quality Board.
- Jensen, A. L., S. A. Spigarelli, and M. M. Thommes. 1982. PCB uptake by five species of fish in Lake Michigan, Green Bay of Lake Michigan, and Cayuga Lake New York. *Can. J. Fish. Aquat. Sci.* (in press)
- Macek, K. J. and S. Korn. 1970. Significance of the food chain in DDT accumulation in fish. *J. Fish. Res. Board Can.* 27: 1496-1498.
- Neely, W. B., D. R. Branson, and G. E. Blau. 1974. Partition coefficient to

measure bioconcentration potential of organic chemicals in fish. Environ. Sci. Technol. 8: 1113-1115.

New York State Department of Environmental Conservation. 1981. Toxic substances in fish and wildlife: 1979 and 1980 Annual Reports. NYDEC Technical Report 81-1(BEP). 138 pp.

Niimi, A. J. and C. Y. Cho. 1981. Elimination of hexachlorobenzene (HCB) by rainbow trout (Salmo gairdneri) and an examination of its kinetics in Lake Ontario salmonids. Can. J. Fish. Aquat. Sci. 38: 1350-1356.

Norstrom, R. J., A. E. McKinnon, and A. S. W. deFreitas. 1976. A bioenergetics-based model for pollutant accumulation by fish. Simulation of PCB and methylmercury residue levels in Ottawa River yellow perch (Perca flavescens). J. Fish. Res. Board Can. 33: 248-267.

Palheimo, J. E. and L. M. Dickie. 1966. Food and growth of fishes. II. Effect of food and temperature on the relation between metabolism and body weight. J. Fish. Res. Board Can. 23: 869-908.

Reinert, R. E., L. J. Stone, and H. L. Bergman. 1974. Dieldrin and DDT: Accumulation from water and food by lake trout (Salvelinus namaycush) in the laboratory. Proc. 17th Conf. Great Lakes Res., pp. 52-58.

Rice, C. Personal communication. Great Lakes Research Division, University of Michigan, Ann Arbor, Michigan.

Sheffy, T. B. 1977. Status report on the PCB problem in Wisconsin. Wisconsin Department of Natural Resources, Madison, Wisconsin. 37 pp.

Sheffy, T. B. and J. R. St. Amant. 1980. Toxic substances survey of Lakes Michigan, Superior and tributary streams; first annual report. Wisconsin Department of Natural Resources, Madison, Wisconsin.

Spigarelli, S. A., M. M. Thommes, and W. Prepejchal. Thermal and metabolic factors affecting PCB uptake by adult brown trout. (submitted for

publication)

- Torrey, M. S. 1976. Environmental status of the Lake Michigan region. Vol. III. Chemistry of Lake Michigan. Argonne National Laboratory Report, ANL/ES-40, Volume 3.
- Thomann, R. V. 1981. Equilibrium model of fate of microcontaminants in diverse aquatic food chains. Can. J. Fish. Aquat. Sci. 38: 280-296.
- Veith, C. D., D. L. Defoe, and B. V. Bergstedt. 1979. Measuring and estimating the bioconcentration factor of chemicals in fish. J. Fish. Res. Board Can. 36: 1040-1048.
- Weininger, D. 1978. Accumulation of PCBs by lake trout in Lake Michigan. Ph.D. Thesis, Water Chemistry, University of Wisconsin, Madison, Wisconsin.

Table 1. Species-contaminant cases used to test the bioenergetics-uptake model.

Species	PCBs	Mirex	Hexachlorobenzene	Dieldrin
Lake trout (<u>Salvelinus namaycush</u>)	Lake Michigan Green Bay Cayuga Lake			Lake Michigan
Brown trout (<u>Salmo trutta</u>)	Lake Michigan Lake Ontario	Lake Ontario	Lake Ontario	
Coho salmon (<u>Oncorhynchus kisutch</u>)	Lake Michigan			
Chinook salmon (<u>O. tshawytscha</u>)	Lake Michigan			
Lake whitefish (<u>Coregonus clupeaformis</u>)	Lake Michigan Green Bay			
Northern pike (<u>Esox lucius</u>)	Lake Michigan			
Burbot (<u>Lota lota</u>)	Lake Michigan			
Carp (<u>Cyprinus carpio</u>)	Lake Michigan Mississippi River			
Channel catfish (<u>Ictalurus punctatus</u>)	Lake St. Clair			
American eel (<u>Anguilla rostrata</u>)			Lake Ontario	Lake Ontario
Smallmouth bass (<u>Micropterus dolomieu</u>)	Lake Ontario	Lake Ontario		

Table 2. Description of parameters and estimates given by Norstrom et al. (1976).

Symbol	Description	Units	Value
e_{cf}	efficiency of assimilation of metabolizable energy from food	-	0.82
e_f	efficiency of assimilation of toxicant from food	-	0.80
e_w	efficiency of assimilation of toxicant from water	-	0.75
e_{ox}	efficiency of assimilation of oxygen from water	-	0.75
β	coefficient relating growth rate to energy associated with growth	-	1.0
γ	exponent of body weight for metabolism	-	0.81
C_{ox}	concentration of oxygen in the water	g/g	9.20×10^{-6}
q_{ox}	caloric equivalent of oxygen	Kcal·g ⁻¹	3.42
k_u	clearance coefficient	g ^{-ξ·wk⁻¹}	0.238
ξ	exponent of body weight in clearance equation	-	0.58
α	low routine metabolism	Kcal·wk ⁻¹ g ^{-γ}	0.12
W_{∞}	asymptotic weight	g	
K	growth coefficient	wk ⁻¹	
b	parameter of length-weight equation	-	
C_f	toxicant concentration in food	mg/kg	
C_w	toxicant concentration in water	mg/kg	
q_f	caloric equivalent of fish	Kcal/g	
q_{fd}	caloric equivalent of food	Kcal/g	

Table 3. Estimated proportions of chemical uptake from water by various species. Uptake from water estimated by setting $C_f = 0$, and projecting uptake. Proportion calculated as quotient of average maximum concentrations.

Species	Lake	PCBs	Dieldrin	Mirex	HCB
Lake trout	Lake Michigan	0.075	0.56		
Salmon	Lake Michigan	0.075			
Brown trout	Lake Michigan	0.10			
Carp	Lake Michigan	0.10			
Whitefish	Lake Michigan	0.10			
Northern pike	Lake Michigan	0.12			
Burbot	Lake Michigan	0.17			
Smallmouth bass	Lake Ontario	0.009		0.020	
Brown trout	Lake Ontario	0.025		0.097	0.22
American eel	Lake Ontario		0.95		0.56
Channel catfish	Lake St. Clair	0.021			
Carp	Mississippi River	0.028			

Table 4. Estimated average maximum concentrations (ppm) of selected chemicals in fish and water of Lake Michigan and calculated bioaccumulation factors (log BAF). Concentration data from Torrey (1976), Sheffy (1977), Sheffy and St. Amant (1980), GLECS (1979), Rice (1982).

Species	PCBs	DDT	Dieldrin	Chlordane	Heptachlor	Toxaphene	HCB
Lake trout	30	6	0.3	1.5	0.8	8	0.5
log BAF	7.18	6.78	5.48	6.70	6.20	6.90	7.00
Brown trout	6			0.4			
log BAF	6.48			6.12			
Coho salmon	8	2	0.3				
log BAF	6.60	6.30	5.48				
Chinook salmon	15	3	0.3	0.6			
log BAF	6.88	6.48	5.48	6.30			
Whitefish	3	2	0.2		0.3		0.1
log BAF	6.18	6.30	5.30		5.78		6.30
Bloater	4	2	0.3		0.3	4	0.2
log BAF	6.30	6.30	5.48		5.78	6.60	6.60
Burbot	2.5	0.2	0.05				
log BAF	6.10	5.30	4.70				
Carp	40	5	1				
log BAF	7.30	6.70	6.00				
Sucker	4		0.2				
log BAF	6.30		5.30				
Yellow perch	3	0.2	0.05				
log BAF	6.18	5.30	4.70				
Walleye	2	0.2	0.05				
log BAF	6.00	5.30	4.70				
Northern pike	3	0.2		0.1			
log BAF	6.18	5.30		5.52			
Alewife	4						
log BAF	6.30						
Smelt	2	0.4	0.05				
log BAF	6.00	5.60	4.70				
H ₂ O, ppt	2.0	1.0	1.0	0.3	0.5	1.0	0.05
log P (octanol/H ₂ O)	6.5	5.8	4.4	6.0	5.4	6.4	6.2

Table 5. Estimated average maximum concentrations of selected chemicals in fish (ppm) and water (ppt) of Lake Ontario and calculated bioaccumulation factors (log BAF). Concentration data from NYDEC (1981) and IJC (1977).

Species	PCBs	DDT	Dieldrin	HCB	Mirex
Lake trout	20	4	0.2	0.3	1
log BAF	7.00	6.60	5.30	6.78	7.30
Brown trout	5	1	0.1	0.07	0.2
log BAF	6.40	6.00	5.00	6.15	6.60
Rainbow trout	4	1	0.05	0.05	0.2
log BAF	6.30	6.00	4.70	6.00	6.60
Coho salmon	8	2	0.1	0.1	0.5
log BAF	6.60	6.30	5.00	6.30	7.00
Chinook salmon	10	2	0.1	0.1	0.5
log BAF	6.70	6.30	5.00	6.30	7.00
Carp				0.2	
log BAF				6.60	
Yellow perch	3	0.2			0.02
log BAF	6.18	5.30			5.60
Walleye	3	0.5	0.02		0.01
log BAF	6.18	5.70	4.30		5.30
Northern pike	4	0.8	0.1		0.05
log BAF	6.30	5.90	5.00		6.00
Smallmouth bass	10	1		0.08	1.0
log BAF	6.70	6.00		6.20	7.30
Alewife	3	0.8			0.07
log BAF	6.18	5.90			6.15
Smelt	3	0.5	0.03		0.07
log BAF	6.18	5.70	4.48		6.15
American eel	10	1.5	0.4	0.1	1.5
log BAF	6.70	6.18	5.60	6.30	7.48
Channel catfish	4	0.8	0.05	0.05	0.3
log BAF	6.30	5.90	4.70	6.00	6.78
H ₂ O, ppt	2.0	1.0	1.0	0.05	0.05
log P (octanol/H ₂ O)	6.5	5.8	4.4	6.2	6.9

Table 6. Statistics for linear regressions of log BAP versus log P for each species. NS = slope not significantly different from zero ($P > 0.05$).

	Lake Michigan				Lake Ontario			
	n	Intercept	Slope	R ²	n	Intercept	Slope	R ²
Alewife	1				3		NS	
Bloater	6	3.14	0.525	0.84				
Brown trout	2	1.80	0.720	1.00	5	2.19	0.644	0.99
Burbot	3		NS					
Chinook salmon	4	2.67	0.637	0.95	5	1.58	0.786	0.98
Coho salmon	3	3.12	0.541	0.99	5	1.64	0.771	0.98
Carp	3	3.32	0.602	0.98				
Lake trout	7	2.03	0.787	0.96	5	1.84	0.798	0.99
Northern pike	3	-2.08	1.27	0.99	4		NS	
Rainbow trout	1				5	1.47	0.746	0.97
Smelt	3	1.97	0.622	0.99	4	1.44	0.709	0.96
Sucker	2	3.20	0.476	1.00				
Walleye	3	2.04	0.592	0.95	4	2.13	0.549	0.57
Whitefish	5	3.23	0.484	0.84				
Yellow perch	3		NS		3		NS	
American eel					5	2.52	0.660	0.83
Channel catfish					5	1.19	0.796	0.98
Smallmouth bass					4	-1.17	1.22	0.95

Table 7. Statistics for linear regressions of log BAF versus % fat for each chemical. NS = slope not significantly different from zero ($P > 0.05$).

log P	Lake Michigan				Lake Ontario			
	n	Intercept	Slope	R ²	n	Intercept	Slope	R ²
4.4 (Dieldrin)	11	4.84	0.029	0.52	10	4.46	0.029	0.53
5.4 (Heptachlor)	3		NS					
5.8 (pp'DDT)	11	5.39	0.045	0.68	13	5.70	0.021	0.35
6.0 (Chlordane)	5		NS					
6.2 (HCB)	3		NS		8		NS	
6.5 (PCBs)	14	6.15	0.022	0.29	13	6.24	0.015	0.27
6.9 (Mirex)					13	5.86	0.052	0.50

LAKE TROUT - LAKE MICHIGAN PCBs

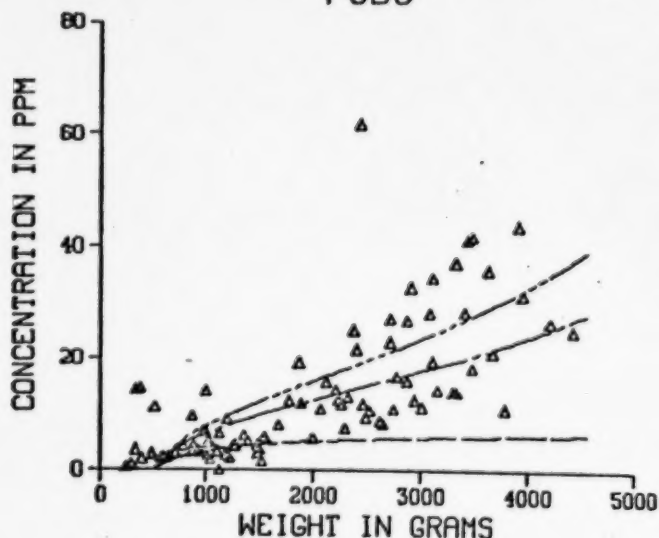


Figure 1. Observed PCB concentrations (Sheffy, 1977) and projections with the model. Growth: $\log_{10} W = -5.39 + 3.16 \log_{10} L$ (Carlander, 1969); $W_{\infty} = 7343$; $K = -0.14$; Metabolic: $\alpha = 0.12$; $\gamma = 1.25, 1.20, 0.81$ (top down); Uptake-elimination: $C_w = 10.0$ ppt; $C_f = 3.0$ ppm; $e_f = 0.80$; $e_w = 0.75$; $\kappa_u = 0.238$. (All other parameter estimates from Norstrom et al., 1976.)

LAKE TROUT - GREEN BAY PCBs

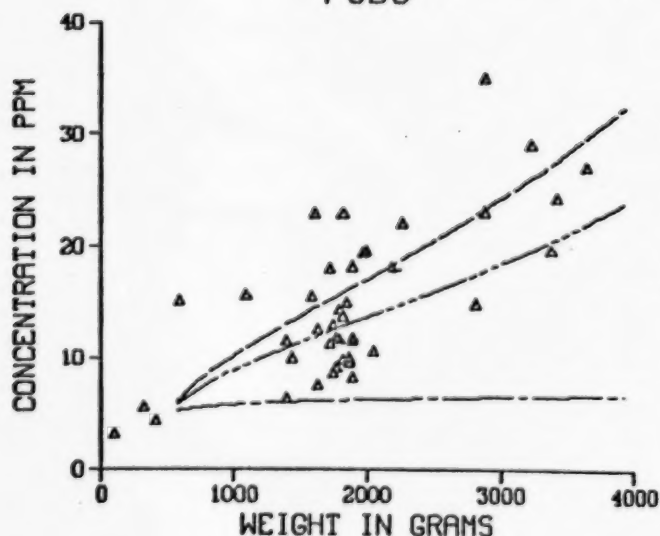


Figure 2. Observed PCB concentrations (Sheffy, 1977) and projections with the model. Growth: $\log_{10} W = -5.39 + 3.16 \log_{10} L$ (Carlander, 1969); $W_{\infty} = 7343$; $K = -0.14$. Metabolic: $\alpha = 0.12$; $\gamma = 1.25, 1.20, 0.81$ (top down). Uptake-elimination: $C_w = 10$ ppt; $C_f = 3.0$ ppm. (All other parameter estimates from Norstrom et al., 1976.)

COHO SALMON - LAKE MICHIGAN PCBs

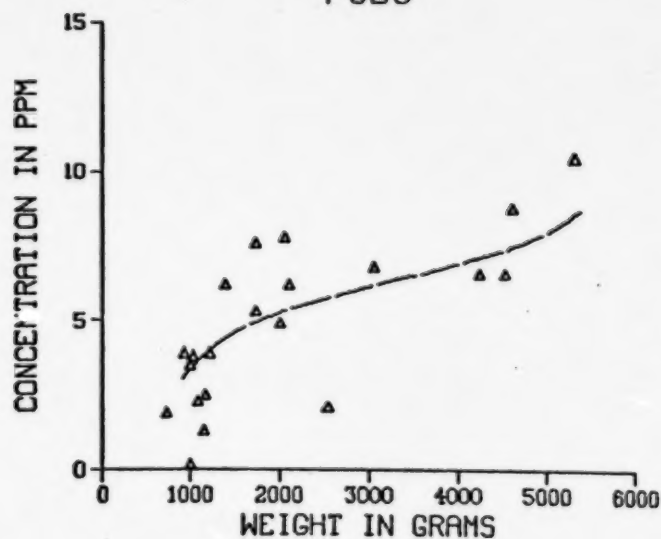


Figure 3. Observed PCB concentrations (Sheffy, 1977) and the projection with the model. Growth: $\log_{10} W = -4.57 + 2.84 \log_{10} L$ (Daley et al., 1975); $W_{\infty} = 5270$; $K = -0.58$. Metabolic: $\alpha = 0.12$; $\gamma = 1.0$. Uptake-elimination: $C_w = 10$ ppt; $C_f = 3$ ppm. (All other parameter estimates from Norstrom et al., 1976.)

CHINOOK SALMON - LAKE MICHIGAN PCBs

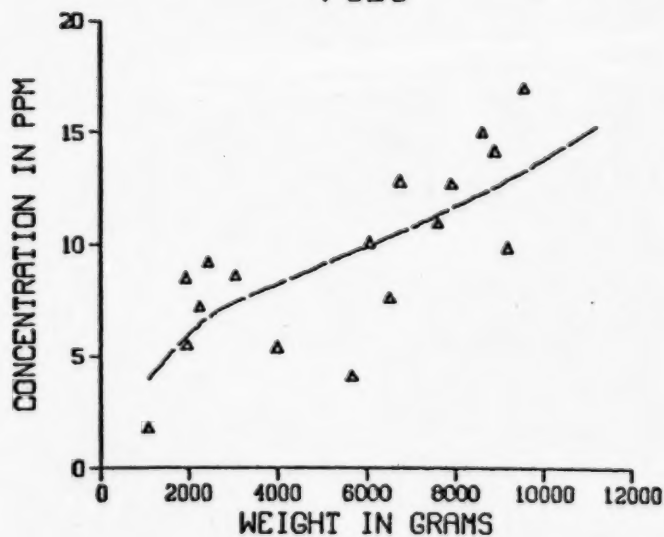


Figure 4. Observed PCB concentrations (Sheffy, 1977) and the projection with the model. Growth: $\log_{10} W = -4.86 + 2.97 \log_{10} L$ (Daley et al., 1975); $W_{\infty} = 18,580$; $K = -0.39$. Metabolic: $\alpha = 0.12$; $\gamma = 1.20$. Uptake-elimination: $C_w = 10$ ppt; $C_f = 3$ ppm. (All other parameter estimates from Norstrom et al., 1976.)

BROWN TROUT - LAKE MICHIGAN PCBs

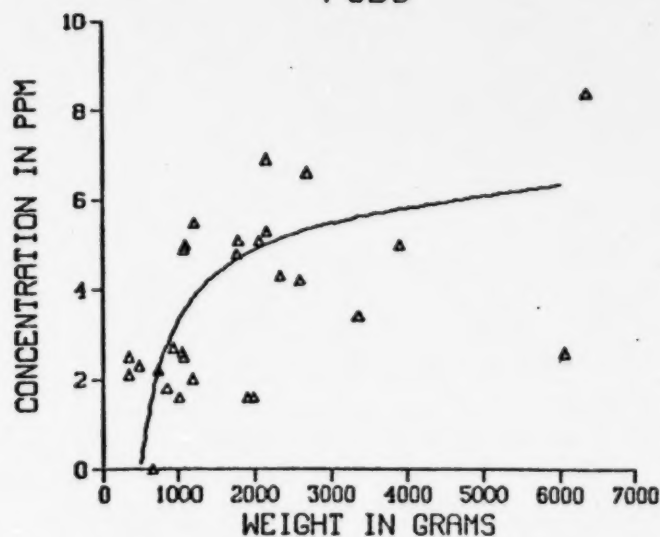


Figure 5. Observed PCB concentrations (Sheffy, 1977) and the projection with the model. Growth: $\log_{10} W = -4.46 + 2.86 \log_{10} L$ (Carlander, 1969); $W_{\infty} = 7760$; $K = -0.23$. Metabolic: $\alpha = 0.12$; $\gamma = 0.81$. Uptake-elimination: $C_w = 10$ ppt; $C_f = 3$ ppm. (All other parameter estimates from Norstrom et al., 1976.)

BURBOT - GREEN BAY PCBs

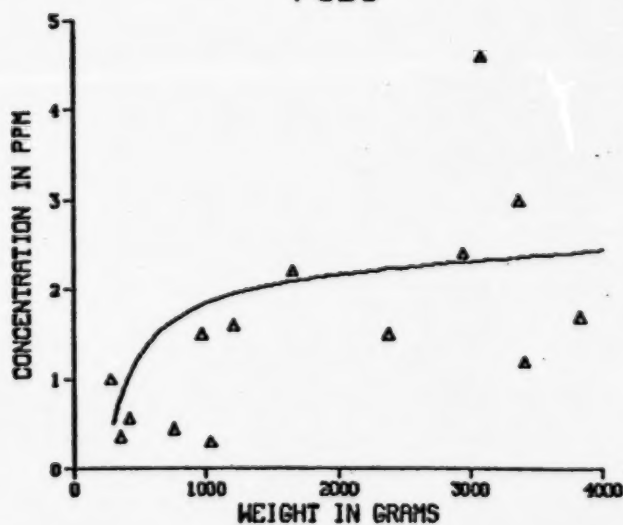


Figure 6. Observed PCB concentrations (Sheffy, 1977) and the projection with the model. Growth: $\log_{10} W = -4.90 + 3.0 \log_{10} L$ (Carlander, 1969); $W_{\infty} = 6000$; $K = -0.16$. Metabolic: $\alpha = 0.12$; $\gamma = 0.80$. Uptake-elimination: $C_w = 10$ ppt; $C_f = 1$ ppm. (All other parameter estimates from Norstrom et al., 1976.)

N. PIKE - GREEN BAY PCBs

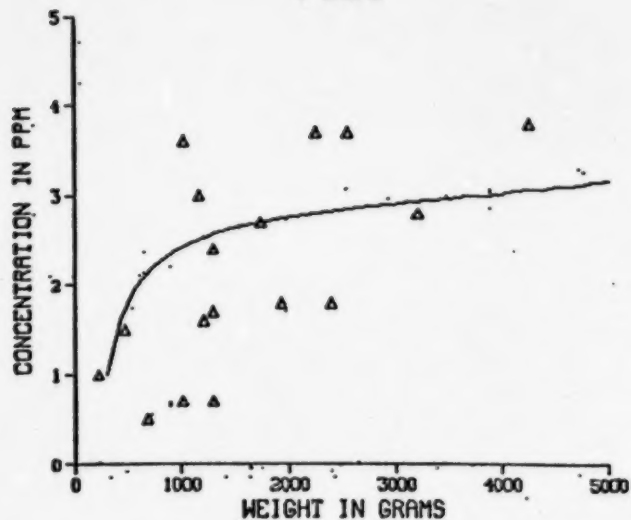


Figure 7. Observed PCB concentrations (Sheffy, 1977) and the projection with the model. Growth: $\log_{10} W = -5.20 + 3.0 \log_{10} L$ (Carlander, 1969); $W_{\infty} = 6300$; $K = -0.30$. Metabolic: $\alpha = 0.15$; $\gamma = 0.80$; $q_{fd} = 1.5$. Uptake-elimination: $C_w = 10$ ppt; $C_f = 2$ ppm. (All other parameter estimates from Norstrom et al., 1976.)

WHITEFISH - LAKE MICHIGAN PCBs

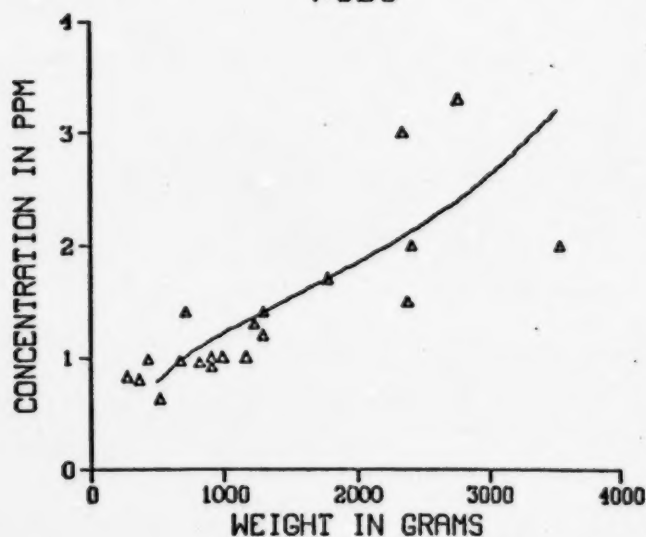


Figure 8. Observed PCB concentrations (Sheffy, 1977) and the projection with the model. Growth: $\log_{10} W = -5.98 + 3.36 \log_{10} L$ (Carlander, 1969); $W_{\infty} = 6000$; $K = -0.30$. Metabolic: $\alpha = 0.12$; $\gamma = 1.20$. Uptake-elimination: $C_w = 10$ ppt; $C_f = 2$ ppm. (All other parameter estimates from Norstrom et al., 1976.)

WHITEFISH - GREEN BAY PCBs

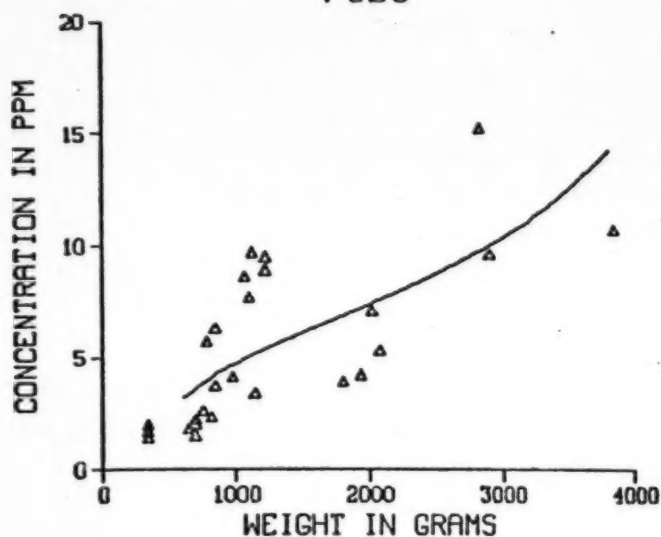


Figure 9. Observed PCB concentrations (Sheffy, 1977) and the projection with the model. Growth: $\log_{10} W = -6.03 + 3.38 \log_{10} L$ (Carlander, 1969); $W_{\infty} = 6000$; $K = -0.30$. Metabolic: $\alpha = 0.12$; $\gamma = 1.20$. Uptake-elimination: $C_w = 10$ ppt; $C_f = 2$ ppm. (All other parameter estimates from Norstrom et al., 1976.)

CARP - GREEN BAY PCBs

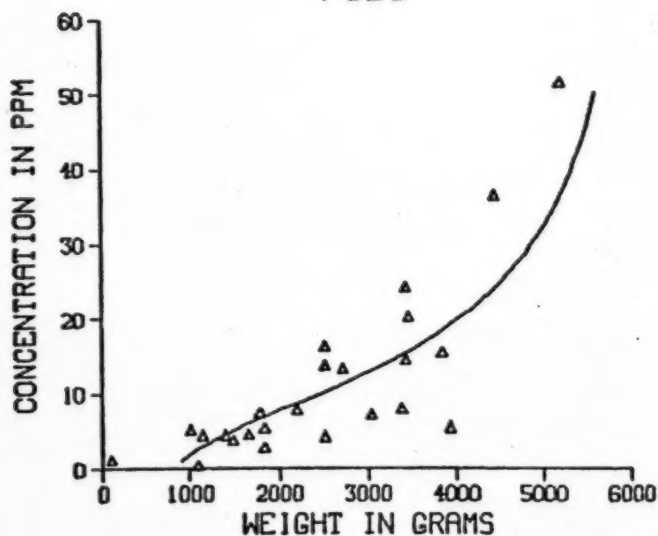


Figure 10. Observed PCB concentrations (Sheffy, 1977) and the projection with the model. Growth: $\log_{10} W = -4.90 + 3.0 \log_{10} L$ (Carlander, 1969); $W_{\infty} = 6000$; $K = -0.30$. Metabolic: $\alpha = 0.20$; $\gamma = 1.30$; $q_f = 1.75$. Uptake-elimination: $C_w = 10$ ppt; $C_f = 1$ ppm.

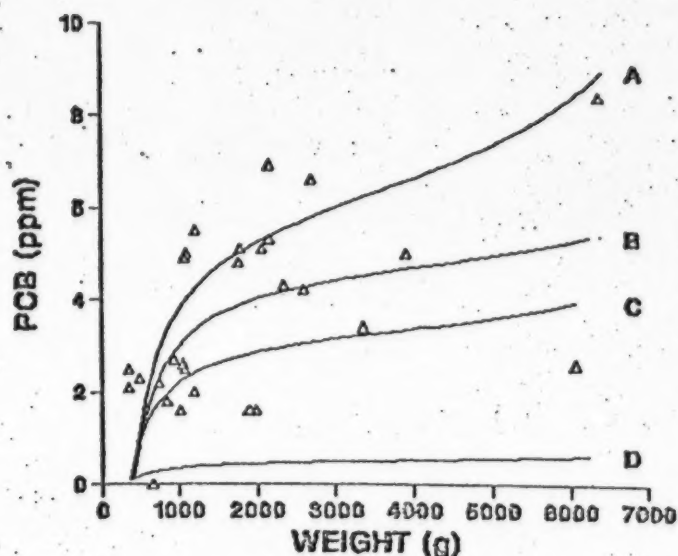


Figure 11. Observed PCB concentrations in brown trout from Lake Michigan (Sheffy, 1977) and projections under four conditions: A) continuous exposure to cyclic temperature regime ($C_f = 2.5$ ppm; $K = -0.23$; $\gamma = 1.0$; $q_f = 1.75$, $q_{fd} = 1.75$); B) continuous exposure to ambient temperatures ($C_f = 2.5$ ppm; $K = -0.22$; $\gamma = 0.90$; $q_f = 1.50$; $q_{fd} = 1.75$); C) continuous exposure to constant 13 C ($C_f = 2.5$ ppm; $K = -0.21$; $q_f = 1.0$; $q_{fd} = 1.75$); D) continuous exposure to ambient temperatures ($C_f = 0$).

BROWN TROUT - L. ONTARIO (FALL) PCBs

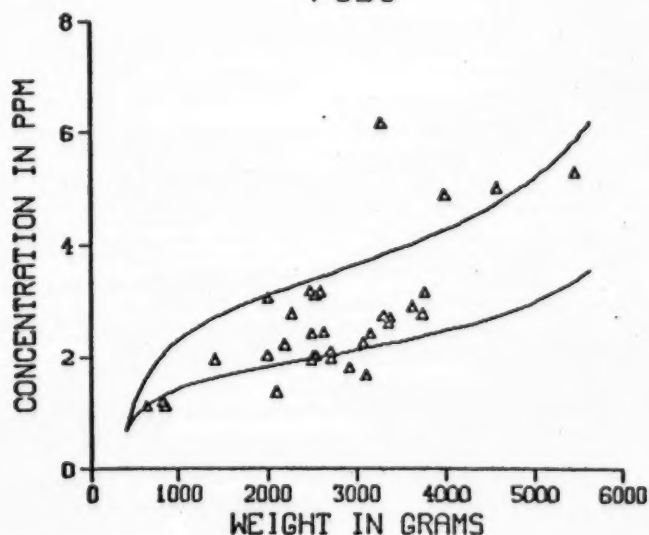


Figure 12. Observed PCB concentrations (NYDEC, 1981) and the projections with the model. Growth: $\log_{10} W = -4.3 + 2.82 \log_{10} L$ (NYDEC, 1981); $W_{\infty} = 6500$; $K = -0.23$. Metabolic: $\alpha = 0.20$; $\gamma = 1.0$; $q_{fd} = 1.0, 1.5$. Uptake-elimination: $C_w = 1$ ppt; $C_f = 2$ ppm, 0.75 ppm.

SM. BASS - L. ONTARIO PCBs

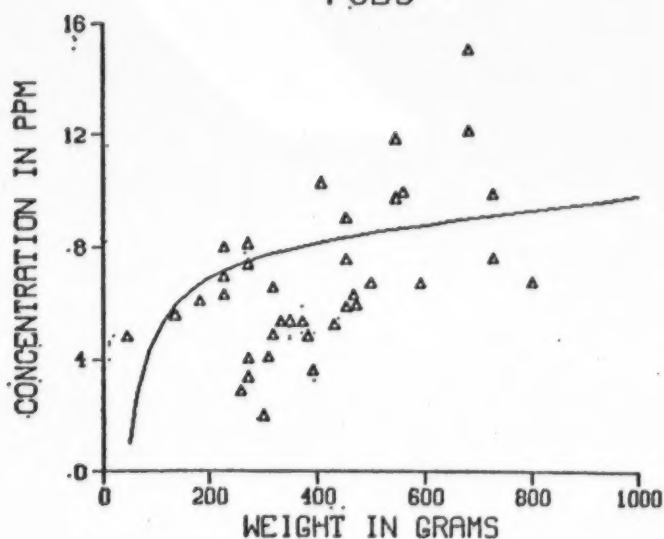


Figure 13. Observed PCB concentrations (NYDEC, 1981) and the projection with the model. Growth: $\log_{10} W = -5.55 + 3.26 \log_{10} L$ (NYDEC, 1981); $W_{\infty} = 2000$; $K = -0.20$. Metabolic: $\alpha = 0.20$; $\gamma = 0.90$. Uptake-elimination: $C_w = 1$ ppt; $C_f = 4$ ppm. (All other parameter estimates from Norstrom et al., 1976.)

CARP - MISSISSIPPI R. PCBs

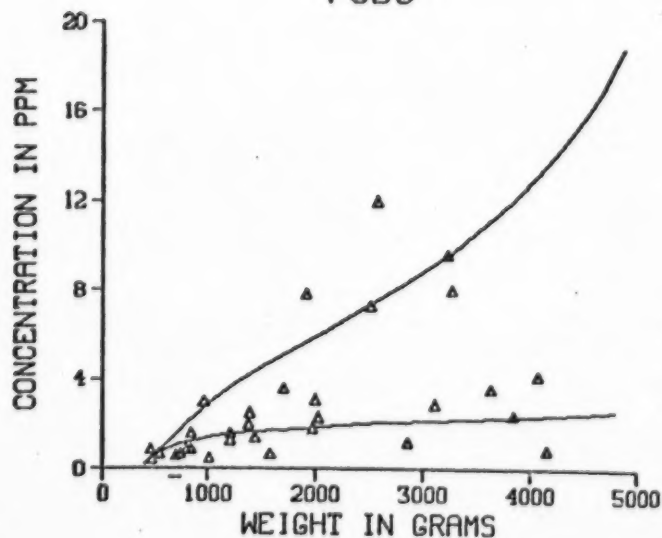


Figure 14. Observed PCB concentrations (Sheffy, 1977) and the projections with the model. Growth: $\log_{10} W = -5.0 + 3.0 \log_{10} L$ (Carlander, 1969); $W_{\infty} = 6000$; $K = -0.30, -0.40$. Metabolic: $\alpha = 0.20$; $\gamma = 0.9, 1.30$; $q_f = 1.0, 1.75$. Uptake-elimination: $C_w = 1$ ppt; $C_f = 1$ ppm. (All other parameter estimates from Norstrom et al., 1976.)

CHANNEL CATFISH - LAKE ST. CLAIR PCBs

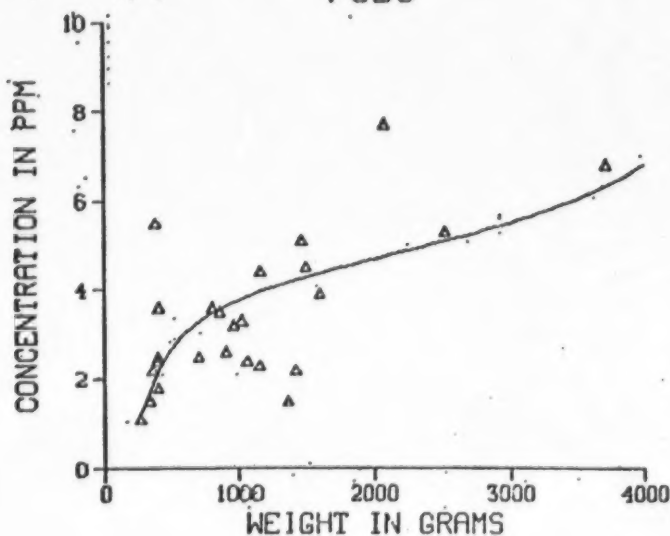


Figure 15. Observed PCB concentrations (GLECS, 1979) and the projection with the model. Growth: $\log_{10} W = -5.95 + 3.35 \log_{10} L$ (GLECS, 1979); $W_{\infty} = 5000$; $K = -0.20$. Metabolic: $\alpha = 0.12$; $\gamma = 1.0$. Uptake-elimination: $C_w = 1$ ppt; $C_f = 2$ ppm. (All other parameter estimates from Norstrom et al., 1976.)

LAKE TROUT - CAYUGA LAKE PCBs

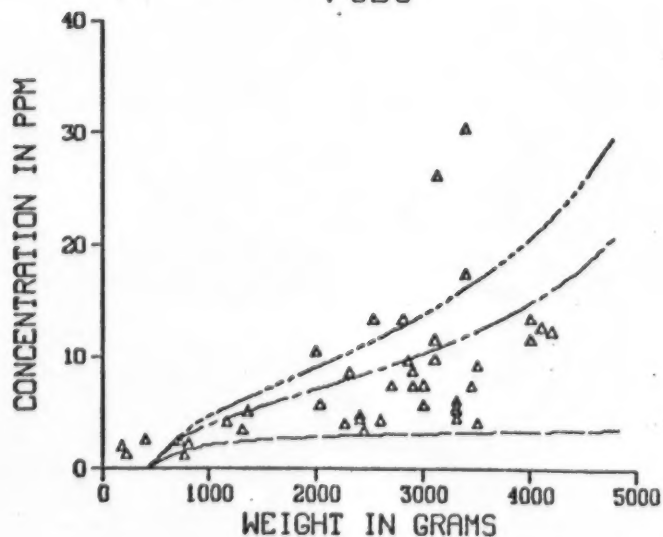


Figure 16. Observed PCB concentrations (Bache et al., 1972) and the projections with the model. Growth: $W_{\infty} = 5940$; $K = -0.16$. Metabolic: $\alpha = 0.12$; $\gamma = 1.25, 1.20, 0.81$ (top down). Uptake-elimination: $C_w = 10$ ppt; $C_f = 2$ ppm. (All other parameter estimates from Norstrom et al., 1976.)

LAKE TROUT - L. MICHIGAN DIELDRIN

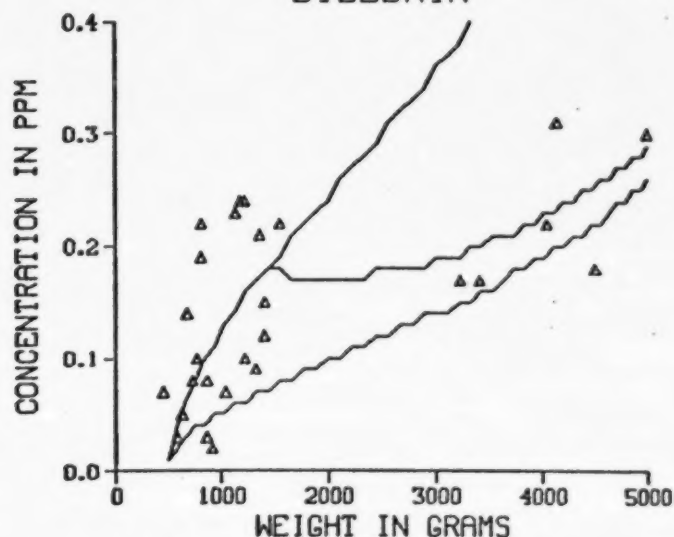


Figure 17. Observed dieldrin concentrations (GLECS, 1979) and the projections with the model. Growth: $\log_{10} W = -5.39 + 3.16 \log_{10} L$ (Carlander, 1969); $W_{\infty} = 7343$; $K = -0.14$. Metabolic: $\alpha = 0.12$; $\gamma = 1.20$. Uptake-elimination: $C_w = 2$ ppt, 0.5 ppt; $C_f = 0.1$ ppm; $e_f = 0.1$; $\kappa_u = 0.60$. (All other parameter estimates from Norstrom et al., 1976.)

A. EEL - L. ONTARIO DIELDRIN

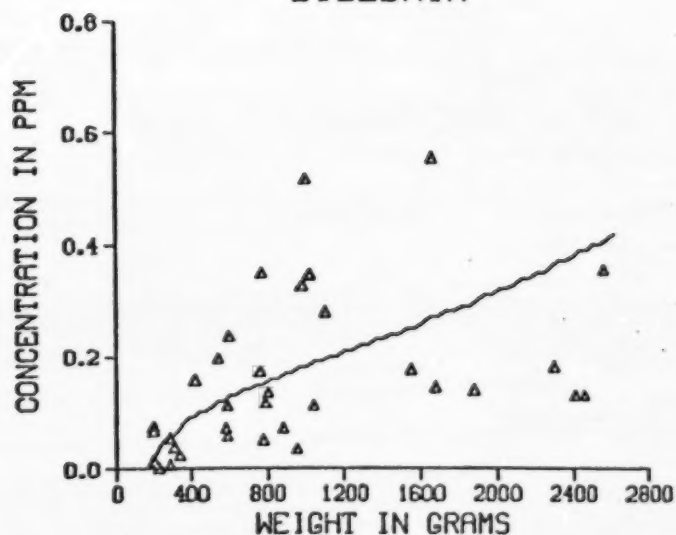


Figure 18. Observed dieldrin concentrations (NYDEC, 1981) and the projection with the model. Growth: $\log_{10} W = -6.50 + 3.20 \log_{10} L$ (NYDEC, 1981); $W_{\infty} = 5000$; $K = -0.07$. Metabolic: $\alpha = 0.15$; $\gamma = 1.20$; $q_f = 2.0$. Uptake-elimination: $C_w = 1$ ppt; $C_f = 10$ ppb; $e_f = 0.10$; $\kappa_u = 0.60$. (All other parameter estimates from Norstrom et al., 1976.)

BROWN TROUT - L. ONTARIO (FALL) HCB

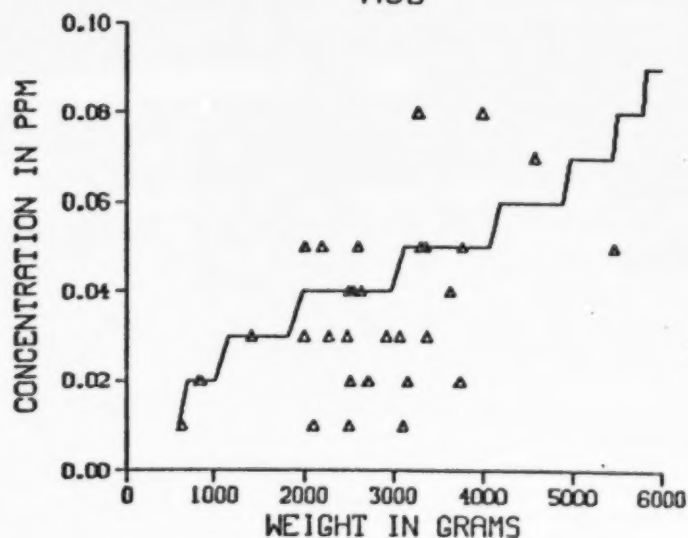


Figure 19. Observed HCB concentrations (NYDEC, 1981) and the projection with the model. Growth: $\log_{10} W = -4.3 + 2.82 \log_{10} L$ (NYDEC, 1981); $W_{\infty} = 6500$; $K = -0.23$. Metabolic: $\alpha = 0.20$; $\gamma = 1.0$. Uptake-elimination: $C_w = 0.1$ ppt; $C_f = 15$ ppb. (All other parameter estimates from Norstrom et al., 1976.)

A. EEL - L. ONTARIO HCB

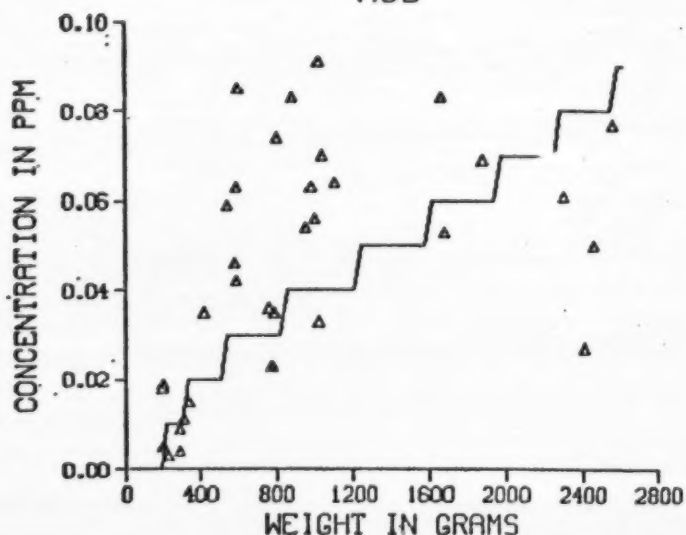


Figure 20. Observed HCB concentrations (NYDEC, 1981) and the projection with the model. Growth: $\log_{10} W = -6.50 + 3.30 \log_{10} L$ (NYDEC, 1981); $W_{\infty} = 5000$; $K = -0.07$. Metabolic: $\alpha = 0.15$; $\gamma = 1.20$; $q_f = 2.0$. Uptake-elimination: $C_w = 0.1$ ppt; $C_f = 3.0$ ppb. (All other parameter estimates from Norstrom et al., 1976.)

BROWN TROUT - L. ONTARIO (FALL) MIREX

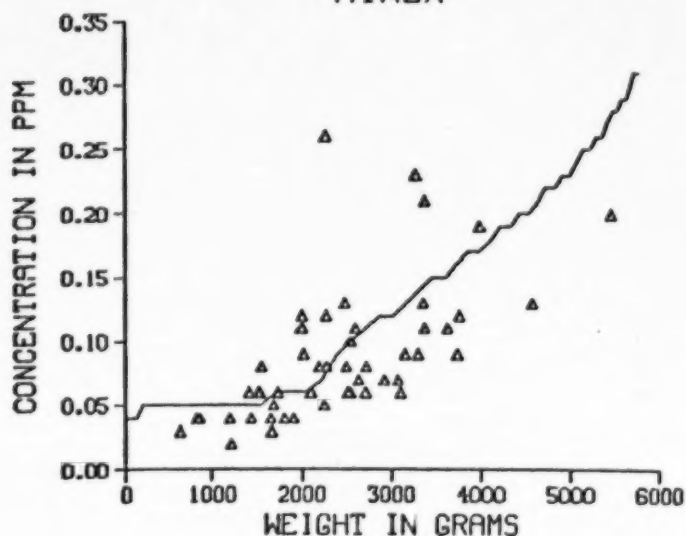


Figure 21. Observed mirex concentrations (NYDEC, 1981) and the projection with the model. Growth: $\log_{10} W = -4.3 + 2.82 \log_{10} L$ (NYDEC, 1981); $W_{\infty} = 6500$; $K = -0.23$. Metabolic: $\alpha = 0.20$; $\gamma = 1.20$. Uptake-elimination: $C_w = 0.1$ ppt; $C_f = 0.02, 0.07$ ppm. (All other parameter estimates from Norstrom et al., 1976.)

SM. BASS - L. ONTARIO MIREX

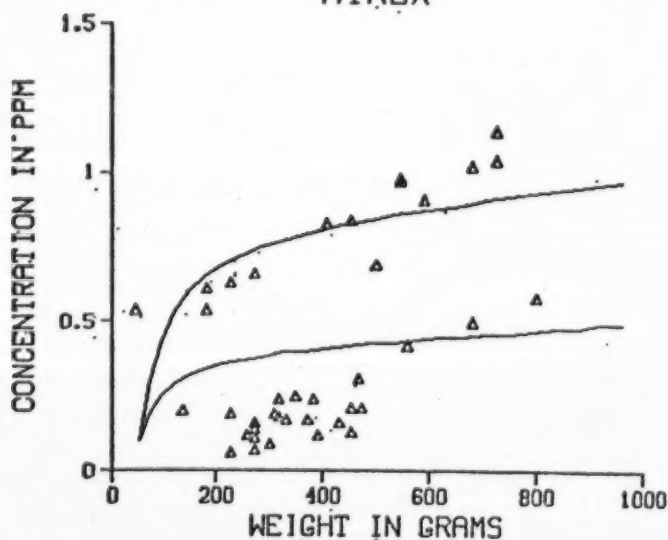


Figure 22. Observed mirex concentrations (NYDEC, 1981) and the projections with the model. Growth: $\log_{10} W = -5.55 + 3.26 \log_{10} L$ (NYDEC, 1981); $W_{\infty} = 2000$; $K = -0.20$. Metabolic: $\alpha = 0.20$; $\gamma = 0.90$. Uptake-elimination: $C_w = 0.1$ ppt; $C_f = 0.2, 0.4$ ppm. (All other parameter estimates from Norstrom et al., 1976.)

LAKE MICHIGAN

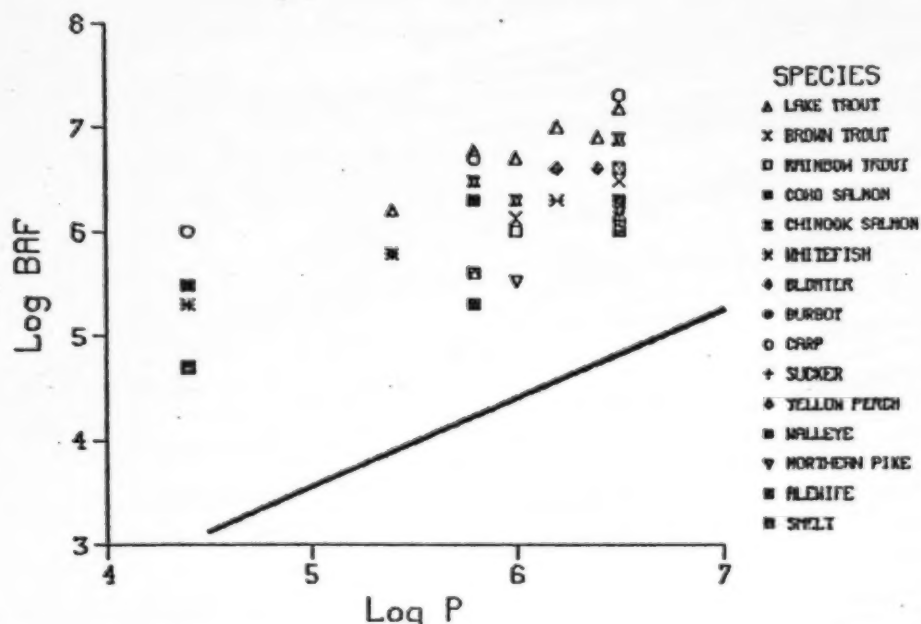


Figure 23. Estimated bioaccumulation factors (log) for fishes in Lake Michigan, as a function of the log P (octanol/water) of chemicals (dieldrin, 4.4; heptachlor, 5.4; pp'DDT, 5.8; chlordane, 6.0; HCB, 6.2; toxaphene, 6.4; PCBs, 6.5). The solid line is the log BCF-log P regression from Veith et al. (1979).

LAKE ONTARIO

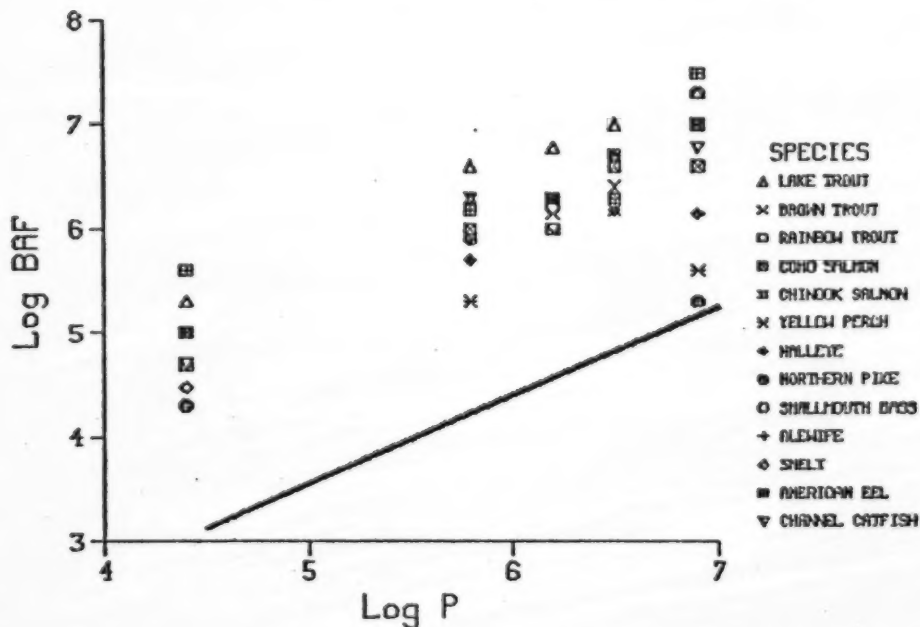


Figure 24. Estimated bioaccumulation factors (log) for fishes in Lake Ontario, as a function of the log P (octanol/water) of chemicals (dieldrin, 4.4; pp'DDT, 5.8; HCB, 6.2; PCBs, 6.5; mirex, 6.9). The solid line is the log BCF-log P regression from Veith et al. (1979).

LAKE MICHIGAN

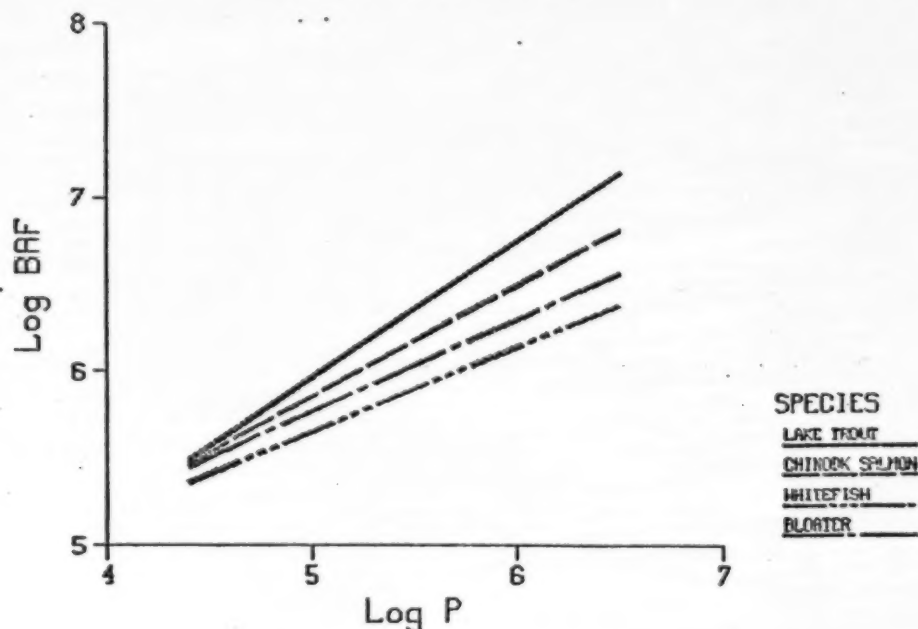


Figure 25. Examples of linear fits to the observed log BAF-log P data for Lake Michigan fishes. Lake trout: $\log \text{BAF} = 2.03 + 0.787 \log P$ (S.E. slope = 0.07; $R^2 = 0.96$; $P < 0.0001$). Chinook salmon: $\log \text{BAF} = 2.67 + 0.637 \log P$ (S.E. slope = 0.106; $R^2 = 0.95$; $P < 0.03$). Whitefish: $\log \text{BAF} = 3.23 + 0.484 \log P$ (S.E. slope = 0.122; $R^2 = 0.84$; $P < 0.03$). Bloater: $\log \text{BAF} = 3.14 + 0.525 \log P$ (S.E. slope = 0.116; $R^2 = 0.84$; $P < 0.02$).

LAKE ONTARIO

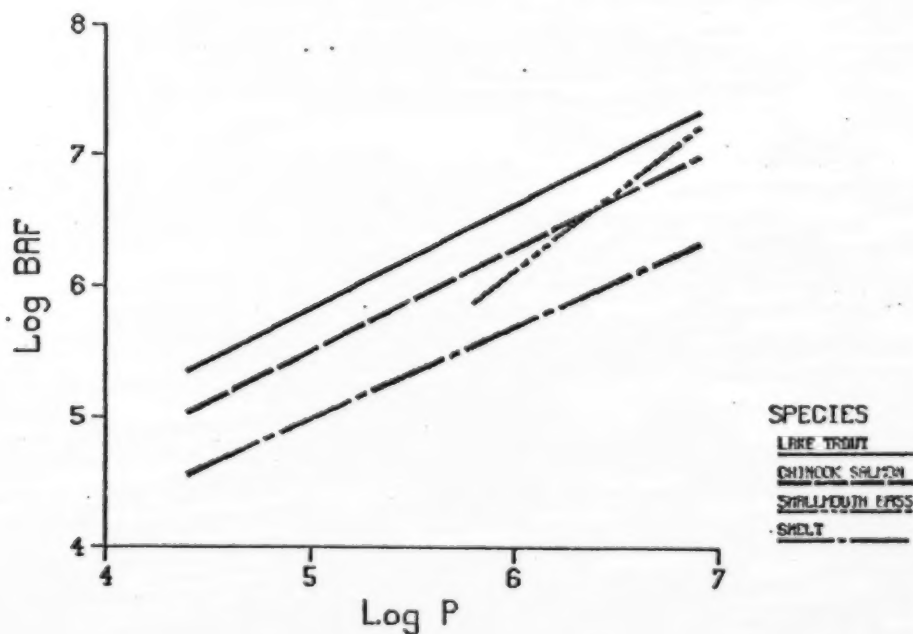


Figure 26. Examples of linear fits to the observed log BAF-log P data for Lake Ontario fishes: Lake trout: $\log \text{BAF} = 1.84 + 0.798 \log P$ (S.E. slope = 0.045; $R^2 = 0.99$; $P < 0.0004$). Chinook salmon: $\log \text{BAF} = 1.57 + 0.786 \log P$ (S.E. slope = 0.068; $R^2 = 0.98$; $P < 0.002$). Smallmouth bass: $\log \text{BAF} = -1.17 + 1.22 \log P$ (S.E. slope = 0.196; $R^2 = 0.95$; $P < 0.03$). Smelt: $\log \text{BAF} = 1.44 + 0.709 \log P$ (S.E. slope = 0.104; $R^2 = 0.96$; $P < 0.03$).

LAKE MICHIGAN

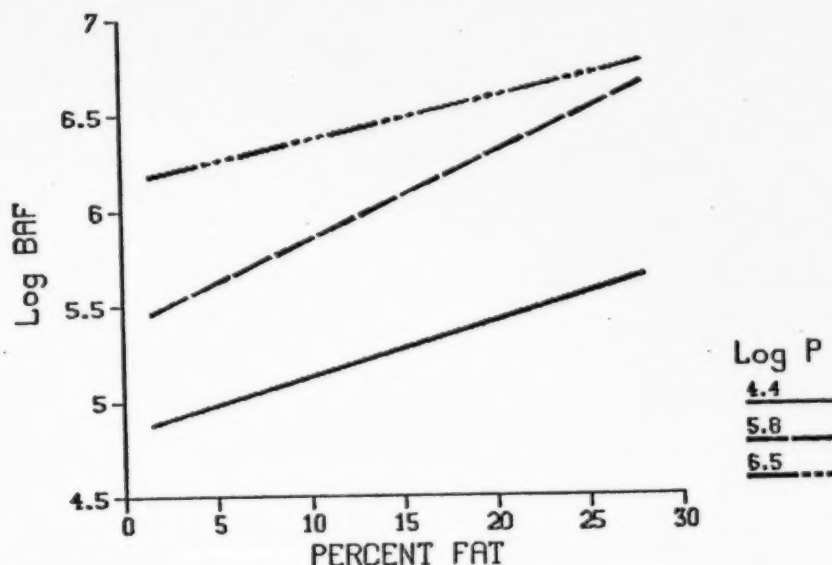


Figure 27. Examples of significant regressions between average maximum percent fat and log BAF for dieldrin (log P = 4.4), pp'DDT (log P = 5.8) and PCBs (log P = 6.5) in Lake Michigan fishes. Dieldrin: log BAF = $4.84 + 0.029 \text{ fat}$ (S.E. slope = 0.009; $R^2 = 0.52$; $P < 0.02$). pp'DDT: log BAF = $5.39 + 0.045 \text{ fat}$ (S.E. slope = 0.010; $R^2 = 0.68$; $P < 0.002$). PCBs: log BAF = $6.15 + 0.022 \text{ fat}$ (S.E. slope = 0.010; $R^2 = 0.29$; $P < 0.05$).

LAKE ONTARIO

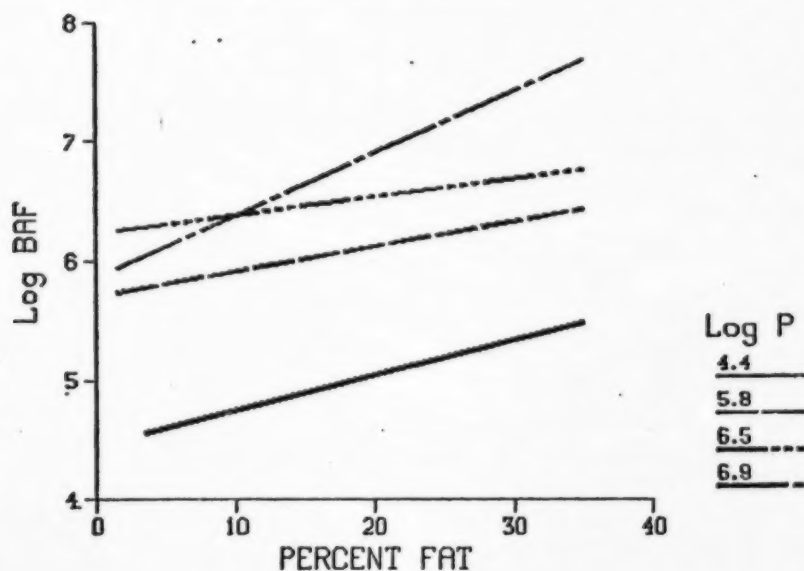


Figure 28. Examples of significant regressions between average maximum percent fat and log BAF for dieldrin (log P = 4.4), pp'DDT (log P = 5.8); PCBs (log P = 6.5); and mirex (log P = 6.9) in Lake Ontario fishes. Dieldrin: log BAF = $4.46 + 0.029 \text{ fat}$ (S.E. slope = 0.010; $R^2 = 0.53$; $P < 0.02$). pp'DDT: log BAF = $5.70 + 0.021 \text{ fat}$ (S.E. slope = 0.009; $R^2 = 0.35$; $P < 0.04$). PCBs: log BAF = $6.24 + 0.015 \text{ fat}$ (S.E. slope = 0.008; $R^2 = 0.27$; $P < 0.05$). Mirex: log BAF = $5.86 + 0.052 \text{ fat}$ (S.E. slope = 0.016; $R^2 = 0.50$; $P < 0.008$).

Several ENVIRONMENTAL RESEARCH CONTRIBUTIONS prepared by the Ecological Sciences Section include:

- 81-15 High abundance and distribution near three heated effluents in Lake Michigan.
- 81-22 Food consumption, growth and lipid deposition by brown trout (Salmo trutta) exposed to constant and fluctuating temperatures.
- 81-23 Estimating growth of brown trout (Salmo trutta) under constant and fluctuating temperatures.
- 81-24 Fish uptake by brown trout (Salmo trutta) and lake trout (Salvelinus namaycush).
- 81-25 A model for the dry deposition of particles to natural water surfaces.
- 81-27 Application of a surplus production model to assess environmental impacts of exploited populations of Daphnia pulex in the laboratory.
- 81-28 The use of box models for estimating dry deposition velocity.
- 81-31 Measurements of the spatial and temporal distributions of chlorophyll and total carbon in southern Lake Michigan by continuous-flow fluorometry and nephelometry.
- 81-32 Ecological compensation and toxicant-induced mortality.
- 81-33 Sorption and sedimentation of Pb and Cd by sediments in southern Lake Michigan.
- 81-34 Bioaccumulation - effects of cadmium and zinc in a Lake Michigan plankton community.
- 81-35 A wind direction controlled air sampling system.
- 81-36 A battery operated data acquisition system.
- 81-37 Determination of cadmium in Lake Michigan by mass spectrometric isotope dilution analysis or atomic absorption spectrophotometry following electrodeposition.
- 81-38 Selected temperatures and thermal experience of brown trout (Salmo trutta) in a steep thermal gradient in nature.
- 81-39 Evidence for the long-distance atmospheric transport of polychlorinated biphenyl (PCB).
- 81-40 Impact of ozone on soybean yield.
- 82-01 Thermal and metabolic factors affecting PCB uptake by brown trout.
- 82-02 Design of an automated, multiposition sampler and selection of an appropriate adsorbent for monitoring atmospheric pollutants.
- 82-03 Cesium-137 in various Chicago foods (collection month - October 1981).
- 82-04 Prediction of chemical accumulation by fish.